Response to Public Comment on Draft 2012 Integrated Report from Brian Sugden of Plum Creek Pertaining to Benthic Algae Thresholds

1.0 Introductory Remarks

The Department is pleased to respond to the comments provided on March 1, 2012 regarding benthic algae thresholds used to determine streams impacts. The commenter stated that the justification for using 120 mg Chl*a*/m^2 may not be applicable to western Montana streams due to their colder temperature regimes and (possibly) different reaeration coefficients. As evidence, the commenter provided water temperature data from western streams and notes that the study from which the 120 mg Chl*a*/m^2 was derived was carried out in a warmer stream located in eastern Montana.

The commenter's concern that the above study occurred "400 miles away" from western Montana is not, in and of itself, necessarily of importance. Studies carried out in different areas of the northern (and southern) hemisphere are often very applicable to western Montana, and elsewhere. For example, Stevenson et al. (2006) recommend a total phosphorous criterion of 30 µg TP/L based on work in streams in Michigan and Kentucky. In fact, their approach, the major type of algae in question (*Cladophora*), and many other factors make the work pertinent to the wadeable streams of Montana. Furthermore, the value they recommend is identical to what was independently derived here (30 µg TP/L) to protect wadeable streams. Similarly, Dodds et al. (1997) extrapolate data from rivers around the world to help develop Clark Fork River nutrient targets in Montana and work in New Zealand (Biggs, 2000) has lead to a recommended threshold there of 120 mg Chl*a*/m^2 to protect streams from nuisance benthic algae.

The main reason the Department is not using a reach average of 120 mg Chl*a*/m^2 as a threshold to assess eastern Montana plains streams is because this algal density is sometimes observed in plains reference streams, whereas it has never been observed in a western MT reference stream (page C-1, Suplee and Sada de Suplee, 2011). Basically, the Department does not wish to set an assessment threshold more stringent than the natural condition of the plains region. In addition, collection and measurement of benthic algae is complicated in plains streams by the frequent presence of macrophytes. Instead the Department is assessing plains wadeable streams using DO deltas, with the caveat that DO deltas may occasionally exceed the recommended threshold due to macrophytes (Suplee and Sada de Suplee, 2011). While diurnal DO monitoring is a more involved and labor intensive approach because it is best done with a deployed instrument, requires time for set-up, deployment, retrieval, and the Department has only a limited number of instruments, it is the preferred option. Collecting benthic algae would be much easier, but our current state of understanding for plains streams suggests that assessment of them via DO delta is a more robust approach.

However, the commenter's concern regarding differing temperatures and reaeration coefficients and extrapolation of findings to western Montana did warrant further investigation. To explore the effect of temperature on the dissolved oxygen (DO) impacts observed in the eastern Montana study, we developed a modified Streeter-Phelps (1925) type analytical model of the Box Elder Creek nutrient dosing reach. The physical basis of the model was then used to evaluate the response of hypothetical streams in western Montana at an elevation of 1,219 m (4,000 ft), mean daily water temperature of 7 ^{o}C (per the provided data), and varying stream characteristics (i.e., reaeration and stream velocities). Methods, discussion and recommendations are provided in **Section 2.0.**

2.0 Simulation of Nutrient Dosing Study Findings for Cooler Water Temperatures, Higher Dissolved Oxygen Saturation, and Differing Reaeration Rates

As documented in Suplee and Sada de Suplee (2011), benthic algae senesced *en masse* at the end of the growing season and led to observed DO levels as low as 1.37 mg/L near the stream bottom in the High Dose reach (HD reach), in a depositional area (glide) with steady laminar flow. It was also documented that the only viable DO sink was the large volume of dead and decaying algae on the stream bottom (confirmed by visual observation and photographs) as water-column BOD samples taken at the time were all less than detection. Thus, the senesced algae are a significant and important DO sink when it comes to eutrophication, and we refer to it here as "senesced algae oxygen demand" (SAOD). The terminology has been used to differentiate it from normal sediment oxygen demand (SOD) which is associated with the oxygen consuming properties of organic material in a stream's bottom sediments.

The questions addressed in this section are:

1. *If the nutrient dosing study had been carried out in an identical western Montana stream but with a cooler temperature and higher dissolved oxygen saturation, would impacts to Montana's dissolved oxygen standards still have occurred?*

2. *Given the stream with cooler water temperature and higher dissolved oxygen saturation simulated in No. 1 above, what would be the effect of differing reaeration rates on the dissolved oxygen?*

Continuously monitored data (by YSI 6600 sonde) at the headwaters of the Box Elder HD reach (just upstream of the nutrient addition point) showed that DO was very close to saturation entering the HD reach (measured value was 8.8 mg DO/L @ 18 cm off the bottom; water temp = 15.3^{o}C; elevation 921 m). By the time water flowed down to the HD reach YSI, DO concentration 18 cm off the bottom had dropped to 1.37 mg/L. The residence time between the two points was about 20 min (as identified through velocity measurements)(**Figure 2-1**). Box Elder Creek is a Rosgen C4 channel (Rosgen, 1996).

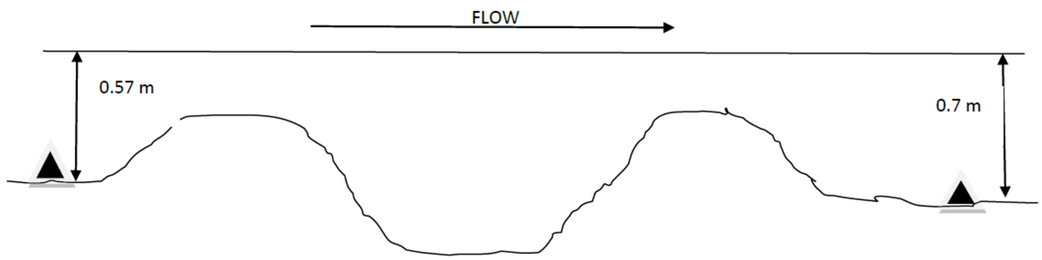

Figure 2-1. Approximate Longitudinal Diagram of Box Elder Creek High Dose Reach. The upstream YSI (black triangle) was placed just upstream of the point where nutrients were added. There is a span of 110 m between it and the High Dose YSI (other black triangle). Two riffles and one pool were found in the reach between the two YSIs.

Data indicate that it is unlikely that 1.37 mg DO/L persisted surface to bottom; rather, a vertical gradient in oxygen concentration occurred. Given the mass of decaying algae, we inferred that DO was zero mg/L at the bottom of the channel whereas, at the surface, it was still probably at or near saturation (as observed upstream). A simple linear relationship was fit between the three DO-depth points (**Figure 2-2**) and, we believe the shape of the curve is very reasonable. In fact, the assumption of a linear gradient is typical of many diffusion problems although it should be noted that a boundary layer with a non-linear gradient towards the bulk fluid would be expected due to turbulent mixing/dispersion (which is different than molecular diffusion that occurs solely from concentration gradients). In this instance, the linear model most reasonably fits the data.

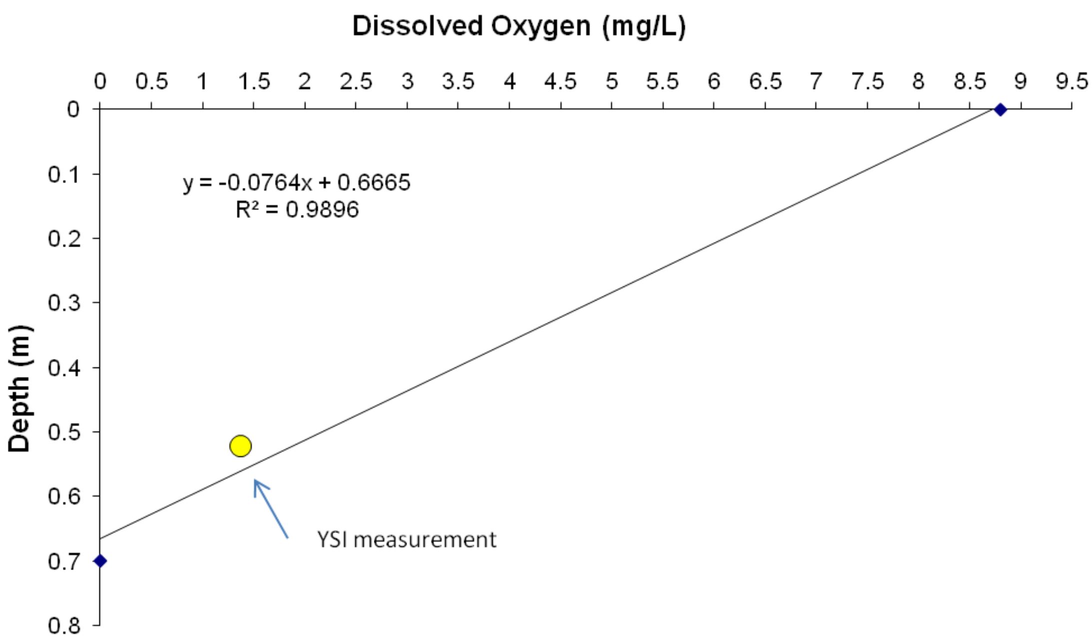

Figure 2-2. Estimated surface to bottom DO profile at the HD reach, 12:01 am, 10/6/2010.

Thus we have some knowledge about the oxygen gradient over the water column as well as the average oxygen concentration at the lower site given the previous assumptions. When using the linear assumption, the midpoint between 0 and 8.8 mg/L is effectively the mean DO concentration at the site (i.e., 4.4 mg/L). To evaluate whether reaeration rates and physiological differences between eastern and western Montana streams do/do not preclude the 120 mg Chl*a*/m^2 algal biomass basis recommended earlier, an analytical solution to Streeter-Phelps model with constant SAOD was developed. Assuming plug flow (i.e., where advection is dominant) and a channel with uniform slope and cross-sectional area, the 1-D mass balance for dissolved oxygen (DO) over a differential element (Δ x) is:

$$\Delta V \frac{DO}{dt} = J_{in} A_c - J_{out} A_c \pm reaction$$

where ΔV = volume of element (m^3)
A_c = channel area (m^2)
B = channel width (m)
H =channel depth (m)
U =channel velocity (m/d)
DO = dissolved oxygen concentration (g/m^3)
J_{in} and J_{out} =flux of DO in and out of element due to advection (g/m^2 d)

Flux in and out of the element is defined as:

$$J_{in} = U(DO)$$
$$J_{out} = U(DO + \frac{\partial c}{\partial x}\Delta x)$$

And by adding first-order reaction rate (i.e., reaeration) and a zero-order term for senesced algae oxygen demand (SAOD [gO_2/m^2 /d]), the advection equation looks like:

$$\Delta V \frac{DO}{dt} = U(DO)A_c - U\left(DO + \frac{\partial DO}{\partial x}\Delta x\right)A_c + k_a\overline{(DO_{sat} - DO}\Delta V - \frac{SAOD}{H}\Delta V$$

Collecting terms, dividing by $\Delta V = A_c \Delta x$, and taking the limit as $\Delta x \to 0$ yields:

$$\frac{DO}{dt} = -U\frac{\partial DO}{\partial x} + k_a(DO_{sat} - DO) - \frac{SAOD}{H}$$

At this point, it should be noted that the first-order reaeration rate k_a is actually a function of the liquid mass transfer velocity [k_l, m/d] divided by *H*, which relates atmospheric flux to surface area. Also, because of the high Henry's constant of O_2, and the fact that oxygen in the atmosphere is constant, exchange is strongly liquid-film controlled. Consequently, DO at saturation (DO_{sat}) can be well characterized by altitude and temperature only. Finally, to simplify the differential equation, we reformulate dissolved oxygen concentration (DO) as dissolved oxygen deficit D=($DO_{sat} - DO$) [mg O_2/L]

, which switches the sign of the DO input/output terms. Under steady state conditions the temporal derivative goes away and we are left with:

$$0 = -U\frac{dD}{dx} - k_a D + \frac{SAOD}{H}$$

The above differential equation can then be solved by using integrating factors and yields the equation below where, D_o=the initial DO deficit (mgO_2/L).

$$D = e^{-\frac{k_a}{U}x}\left[D_o + \frac{SAOD}{Hk_a}\left(e^{\frac{k_a}{U}x} - 1\right)\right]$$

Finally, substitution of k_a with the approximation from Owens et al. (1964) makes the equation appropriate for small streams (Covar, 1976) (in metric units, where U is in m/s):

$$k_a = 5.32\frac{U^{0.67}}{H^{1.85}}$$

Thus the equation is applicable to small shallow streams where the oxygen generation and consumption processes are primarily reaeration and SAOD. It should be noted that reaeration is temperature adjusted using the Arrhenius equation with a theta (θ) of 1.024 (Chapra, 1997). Also, since we have omitted respiration and photosynthesis from our equation, it is appropriate for night-time conditions only. Finally, note that algal respiration does consume oxygen in dark reactions through carbon oxidation, but it was not included in the model.

Following model development, we then calibrated the analytical model to data from Box Elder Creek to estimate SAOD expected from an accumulation of dead/decaying algae (which could then be transferred to other streams). In this instance the calibrated SAOD was very high, approximating 92 gO_2/m^2/day. Using the Arrhenius equation and a theta (θ) of 1.047 (Chapra, 1997), this zero-order rate equates to 76 gO_2/m^2/day at a stream temperature of 7 ^oC. In other words, the biological decomposition rate of the dead algae has been reduced mathematically to that due to the colder water temperature. We refer to this new, adjusted SAOD as $SAOD_{COLD}$.

Figure 2-3 below compares the model output for Box Elder Creek vs. the simulated, carbon-copy western Montana stream (but at 7^o C and an elevation of 1,219 m) evaluated through the model. In general, we see a longitudinal decline in DO concentration reflective of a waterbody flowing over a very large diffuse SAOD source. The zero-order SAOD used for the western MT stream was $SAOD_{COLD}$. The model output presents average water column DO (i.e., the midpoint of the surface-to-bottom curve in **Figure 2-2**) longitudinally, and shows how that average would decline over space due to SAOD. Note that the DO standard is actually exceeded sooner in the western Montana simulation (occurs about 75 m downstream as opposed to 100 m in Box Elder Creek).

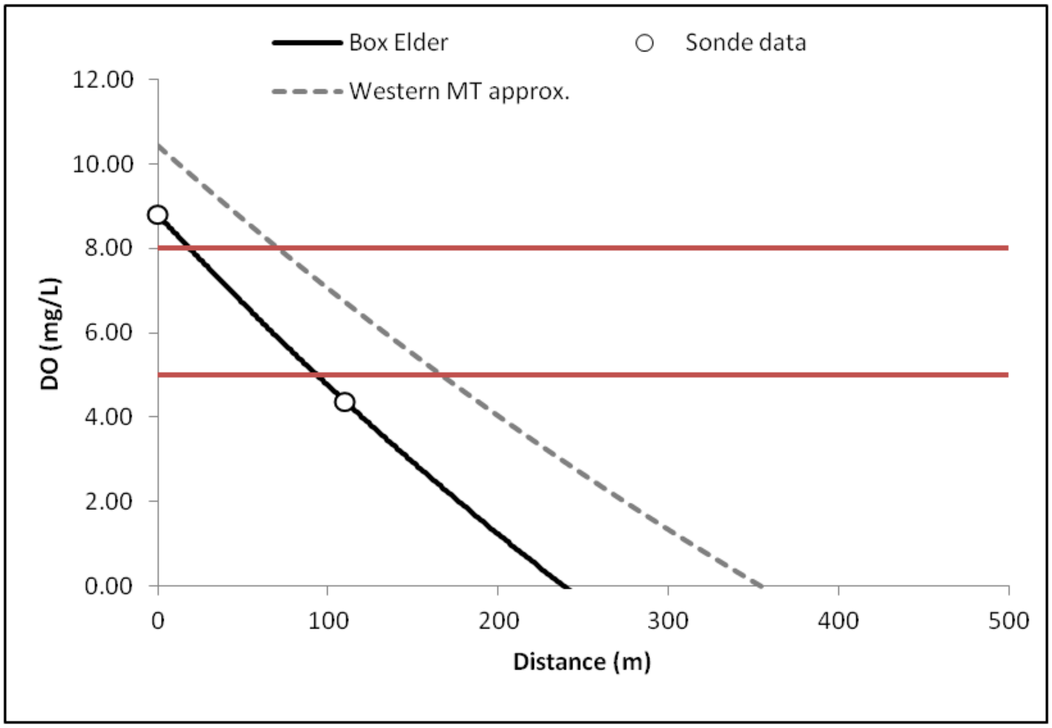

Figure 2-3. Model output for two scenarios: (1) Box Elder Creek nutrient dosing study, including YSI sonde data from the HD reach which were used to calibrate the model, and (2) the simulation of an identical stream at 7^{o} C and at 1,219 m elevation. Red horizontal lines show the DO standards typical for each region. Dissolved oxygen impacts also occur in the colder, higher DO-saturation simulation.

2.1 Simulations Using Different Reaeration Rates

To evaluate the potential effect of the calibrated SAOD mentioned previously, a number of stream configurations similar to these encountered in western Montana were evaluated. The primary difference between these and the Box Elder Creek calibration was the dependence of reaeration rate on channel configuration, and the effect of colder temperatures and altitude on oxygen saturation and biological decomposition. Assumptions made in the western Montana streams were predicated on the Rosgen stream classification system which integrates factors such as slope, channel width/depth ratio, substrate, etc. Given the dependence of the reaeration coefficient on channel depth and velocity, Manning's equation was used to determine effective velocities for various channel configurations:

$$U = 1\frac{1}{n}R^{2/3}S^{1/2}$$

where U = velocity in m/sec, R = hydraulic radius in m, S = water surface slope in m/m (Dunne and Leopold, 1978, but formula from the citation converted to SI units), and n = Manning's coefficient. Rosgen (1996) provides descriptive statistics for many of his stream types (e.g., C4, F4 channels), and values (e.g., surface water slope, cross-sectional area) representing the central tendency of each group were selected and input into Manning's relationship to derive a representative velocity for the stream class group (**Table 2-1**). Roughness coefficients between 0.05 and 0.06 were selected which, from our experience, are reflective of streams during low-flow conditions (recall that the roughness coefficient actually is not independent of flow and depth). Chow (1959) reports variation in Manning's *n* with stage and also that weeds (i.e., macrophytes and attached algae) in stream channels induce somewhat higher Manning's *n* values. Given that our simulated streams would have fairly thick mats of filamentous algae, slightly higher-than-texbook Manning's *n* values are well justified[1].

Table 2-1. Stream Channel Characteristics for Different Representative Rosgen Stream Channels Used in the Model.

Rosgen Stream Type	X-sectional area (m^3)	Width (m)	Depth (m)	Surface Water Slope (m/m)	Manning's *n* used	Velocity (m/sec)
B4	1.54	5.77	0.27	0.0200	0.05	1.11
C3	3.09	11.55	0.27	0.0023	0.06	0.32
C4	2.70	10.09	0.27	0.0045	0.06	0.45
C5	2.51	9.39	0.27	0.0005	0.06	0.14
E3	0.88	1.16	0.76	0.0100	0.06	0.80
E4	0.54	0.71	0.76	0.0100	0.06	0.65
E5	0.56	0.73	0.76	0.0010	0.06	0.21
F4	1.95	7.31	0.27	0.0018	0.06	0.28
G4	0.74	2.78	0.27	0.0200	0.06	0.87

The calculated velocities and depths for the stream groups (**Table 2-1**) were used to recalculate the reaeration coefficient in the model and evaluate the effects on longitudinal DO decline. As before, the simulations were run under the assumption that the stream, irrespective of whether it was C4, B4, etc., was at 7^{o} C and at an elevation of 1,219 m. We tested Rosgen B, C, E, F, and G stream types. Following initial testing, it was clear from the B-channel results that the higher gradient A channels need not be evaluated since their gradients/velocities would overcome SAOD. We did not test D channels as Rosgen (1996) provides no descriptive statistics.

Based on this work, it was found that C and F channels would all be vulnerable to low oxygen problems (i.e., their velocities were insufficient to overcome $SAOD_{COLD}$), the Rosgen C4 channel somewhat less so than C3 or C5. In contrast, A, B, E, and G channels would probably not be impacted by low DO. It is not clear whether D channels would be vulnerable to DO problems or not; some of the lower gradient ones very likely would. **Figure 2-3** below contrasts results from three examples; a hypothetical B4 channel, C3 channel, and F4 channel. As can be seen, high velocities in the B4 channel lead to higher reaeration rates capable of overcoming the $SAOD_{COLD}$; in contrast, the C3 and F4 channel become impacted (relative to their respective DO standard) although at longitudinal distances somewhat extended relative to Box Elder Creek.

[1] Box Elder Creek, which is a Rosgen C4 channel with a D_{50} of 45 mm (coarse gravel), required an even higher Manning's *n* (0.08) in order to match the Manning's equation result to actual mean stream velocity and slopes measured there. Thus, the use of 0.06 in our calculations for Rosgen stream types is quite reasonable.

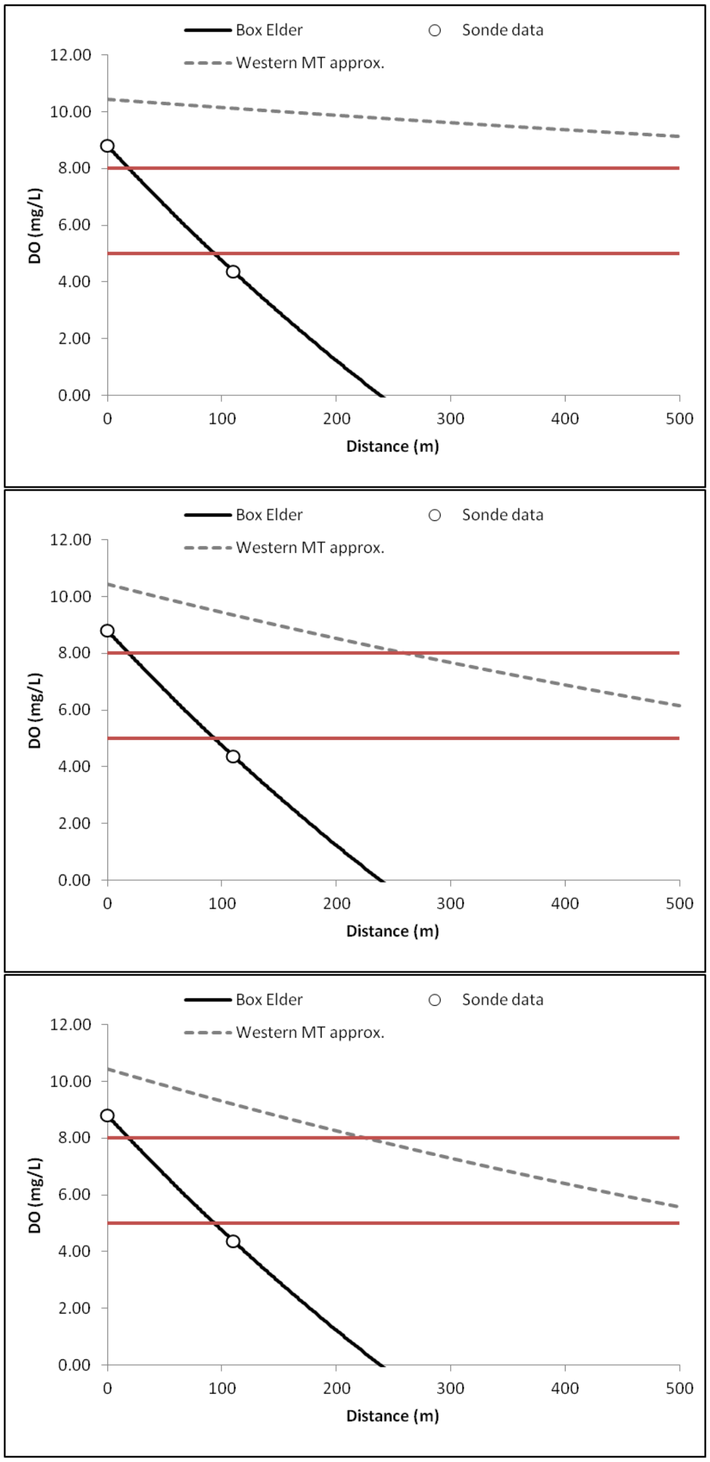

Figure 2-4. Modeled DO impacts as a function of stream class. Upper Panel. Rosgen B4 channel. Dissolved oxygen would not fall below the 8.0 mg DO/L standard for a long distance and DO impacts are almost certainly negated. Middle Panel. Rosgen C3 channel. Dissolved oxygen impacts would occur about 150 m further on longitudinally than what was observed in the Box Elder Cr study. Bottom Panel. Rosgen F4 channel. Dissolved impacts similar to C3 channel, but occur more quickly.

2.1 Discussion and Conclusions for Section 2.0

Others discuss end-of-growing season senescence of plants and its affect on stream water quality. Jewell (1971) notes in streams in England that "At the end of the growing season, or when the weeds are killed, their decomposition may exert heavy demands on the oxygen resources of a water". Novotny and Bendoricchio (1989) observe that "oxygen deficiency is highest and most troublesome in streams where shallow productive zones are followed be deeper sections". The latter statement conforms to what we observed, where senesced algae accumulated in the glides and pools of the Box Elder Creek after it had been dosed with nutrients, and these areas manifested discontinuous areas of low DO longitudinally along the reach.

The level of benthic algae leading to the DO problem at the Box Elder Creek HD reach (127 mg Chl*a*/m^2, 33 g AFDW/m^2) has been observed in eutrophied streams in western Montana, and quite often *Cladophora* provides a substantial proportion of this biomass (just as was observed in the Box Elder study). Thus, the key biological characteristics of the plains stream we studied, in terms of the biomass and algae, are quite comparable to a western Montana stream. Therefore, we believe it can be reasonably concluded that if the study had been carried out in an identical stream in western Montana, but at 7^o C and with DO at saturation of 10.4 mg/L, one would have seen exceedences of the DO standards there as well.

The SAOD we calculated is far higher than sediment oxygen demand (SOD) reported in the literature (highest SOD located was 21.4 g O_2/m^2/day; Ling et al., 2009). But as mentioned at the start, SAOD is not SOD in the normal sense and the rates are not strictly comparable. Novotny and Olem (1994) report that feedlot runoff (a highly organic, putrescible material) can have BOD_5 of 1,000-12,000 mg/L. If this material were all to settle to the stream bottom in a stream having the same average depth as Box Elder Creek and then exert its DO demand, BOD_5 values of this magnitude would equate to 53-640 g O_2/m^2/day (at 20^o C). The SAOD we calculated for decomposing benthic algae (equal to 133 g O_2/m^2/day @ 20^0C) clearly falls to the low side of this range and is, therefore, a realistic estimate. It should also be noted that the true SAOD was probably much higher, but much more localized spatially along the reach (i.e., we calibrated it assuming SAOD over the entirety of the reach).

In conclusion then, it appears that the Box Elder dosing study, had it been carried out under identical circumstances except for lower water temperature and higher DO saturation, would have led to similar conclusions about the impacts of senesced algae oxygen demand and associated effects on stream DO dynamics (**Figures 2-2, 2-3**). However, our simulations indicate that gradients of some western Montana streams are sufficiently high that SAOD can, for all practical purposes, be overcome (**Figure 2-4**, upper panel). Based on our findings, we recommend the following:

1. The 120 mg Chla/m^2 threshold should apply to all Rosgen C and F channels, as their group characteristics (velocity, depth, etc.) appear to be insufficient to overcome DO impacts from senesced algae (SAOD)

2. The 120 mg Chla/m^2 threshold should not apply to Rosgen A, B, E and G channels, as their group characteristics (velocity, depth, etc.) appear to be sufficient to overcome DO impacts from senesced algae (SAOD).
3. No recommendation is made for Rosgen D channels at this time. These channels are not so commonly encountered in Montana and cases will need to be evaluated case-by-case.

The Department agrees with the commenter that, for those streams in which the 120 mg Chl*a*/m^2 should not apply, then the recreationally-derived benthic algal threshold should apply instead.

3.0 Discussion of the Benthic Algal Recreational Criteria Threshold, and the Use of 35 g AFDW/m^2

The Department does not strictly agree with the commenter that 165 mg Chl*a*/m^2 should be the threshold against which measured Chl*a* levels are assessed for the recreational beneficial use. The Department is assuming that in most cases our Chl*a* SOP method will have been the only one used. The simple approach would be to state that all sample averages above 150 mg Chla/m^2 (per Suplee et al., 2009) are impaired. But this does not take into account replicate variation in the original photos or the known variation associated with a sampling method. The approach described in Suplee and Sada de Suplee (2011) accounts for both. As stated therein, "any measured average Chl*a* value DEQ believes could plausibly be as high as 165 mg Chl*a*/m^2 should be considered an exceedence". Based on the known statistical dispersion of our SOP method, that equates to ~130 mg Chla/m^2. And as pointed out in Section B.1.4 of that document's Appendix B, DEQ is willing to consider Chl*a* averages that have been determined using more replicates (eg., 20 replicates), which may change the exceedence threshold. However it should fall to the concerned party to pay for the additional sampling cost <u>and</u> to demonstrate that additional statistical precision was in fact achieved. But again, the Department is assuming that in most cases our SOP method will have been the only one used.

The commenter should also note that 130 mg Chl*a*/m^2 is an uncommonly high average benthic algae density. Dodds et al. (2002) show that, for wadeable stream worldwide in the temperate regions, over 90% of all reported benthic Chla averages are less than 130 mg Chl*a*/m^2. Based on years of sampling in the Clark Fork River and elsewhere around western Montana, the Department has found that 130 mg Chl*a*/m^2 is very unlikely to be encountered in the absence of cultural eutrophication. That value also falls nicely between the two benthic algae standards used on the Clark Fork River (100 and 150 mg Chl*a*/m^2), and is very close to the maximum value used in New Zealand (120 mg Chl*a*/m^2) for aquatic life and recreation (Biggs, 2000).

The Department believes the justification for the use of 35 g AFDW/m^2 in addition to Chl*a* is well supported by work in both Montana and elsewhere; see page B-5 of Appendix B in Suplee and Sada de Suplee (2011). Note in particular that AFDW increased across the growing season at the HD reach in the Box Elder dosing study, ending on a replicate mean of 33 g AFDW/m^2 (Suplee and Sada de Suplee, 2011).

4.0 Conclusion

The Department has given careful consideration to the totality of information provided by the Box Elder nutrient dosing study, modeling results derived thereof, and the recreationally-based benthic Chl*a* value (including consideration of sampling variability). The data show that some western Montana streams would be vulnerable to low DO problems if benthic algae reached 127 mg Chl*a*/m^2 (33 g AFDW/m^2) , and others would not. Further, the Department continues to stand behind the statistical rationale for using ~130 mg Chl*a*/m^2 as the recreational-use threshold (given the statistical dispersion documented for Chl*a* sampling in the Department SOP). At this point, then, there is relatively little distance between the aquatic life/fisheries DO-based Chl*a* threshold and the recreational Chl*a* threshold. Further, monitoring staff who have completed many stream assessments involving benthic algae have indicated that in the vast majority of cases benthic algae levels are either well below or well above the thresholds in question (i.e., borderline cases near the thresholds that would require more detailed analysis and data collection are very uncommon). Thus, In order to preclude a complex, two-threshold system requiring Rosgen stream class identification in all cases, the Department instead advises a single benthic Chl*a* threshold of 125 mg Chl*a*/m^2 (site average), or 35 g AFDW/m^2. Values above these thresholds will be considered impacts to the aquatic life and the recreational beneficial uses.

5.0 References

Biggs, B.J.F., 2000. New Zealand Periphyton Guidelines: Detecting, Monitoring and Managing Enrichment in Streams. Prepared for the New Zealand Ministry of the Environment, Christchurch, 122 p. http://www.mfe.govt.nz/publications/water/periphyton-guideline-dec2000/index.html

Chapra, S.C., 1997. Surface Water Quality Modeling. WCB McGraw-Hill, Boston, Massachusetts, U.S.A.

Chow, V.T., 1959. Open-Channel Hydraulics, McGraw-Hill Book Co, p. 680.

Covar, A.P., 1976. Selecting the Proper Reaeration Coefficient for Use in Water Quality Models. Presented at the U.S. EPA Conference on Environmental Simulation and Modeling. April 19-22. Cincinnati, OH.

DEQ (Montana Department of Environmental Quality), 2010. Circular DEQ-7, Montana Numeric Water Quality Standards, August 2010. http://deq.mt.gov/wqinfo/Standards/default.mcpx

Dodds, W.K, V.H. Smith, and K. Lohman, 2002. Nitrogen and Phosphorus Relationships to Benthic Algal Biomass in Temperate Streams. Canadian Journal of Fisheries and Aquatic Sciences 59: 865-874.

Dunne, T., and L.B. Leopold, 1978. Water in Environmental Planning. W.H. Freeman and Company, New York, U.S.A.

Jewell, W.J., 1971. Aquatic Weed Decay: Dissolved Oxygen Utilization and Nitrogen and Phosphorus Regeneration. Journal of the Water Pollution Control Federation 43: 1457-1467.

Owens, M., Edwards, R., and J. Gibbs, 1964. Some Rearation Studies in Streams. Int. J. Air Water Poll. 8:469-486.

Ling, T, C. Ng, Lee, N., and D. Buda, 2009. Oxygen Demand of the Sediment from the Semariang Batu River, Malaysia. World Applied Sciences Journal 7: 440-447.

Novotny, V., and G. Bendoricchio, 1989. Linking Nonpoint Pollution and Deterioration. Water Environment & Technology 1 (Nov): 400-407.

Novotny, V., and H. Olem, 1994. Water Quality: Prevention, Identification, and Management of Diffuse Pollution. John Wiley and Sons, Inc., New York, U.S.A.

Rosgen, D., 1996. Applied River Morphology. Wildland Hydrology, Pagosa Springs, Colorado, U.S.A.

Stevenson, R. J., S.T. Rier, C.M. Riseng, and R.E. Schultz, 2006. Comparing Effects of Nutrients on Algal Biomass in Streams in Two Regions with Different Disturbance Regimes and with Applications for Developing Nutrient Criteria. Hydrobiologia 561: 149-165.

Streeter, H.W. and E.B. Phelps, 1925. A Study of Pollution and Natural Purification of the Ohio River, III. Factors Concerning the Phenomena of Oxidation and Reaeration. U.S. Public Health Service. Public Health Bulletin No. 146, February, 1925.

Suplee, M.W.,V. Watson, M. Teply, and H. McKee, 2009. How Green is too Green? Public Opinion of What Constitutes Undesirable Algae Levels in Streams. Journal of the American Water Resources Association 45: 123-140.

Suplee, M.W., and R. Sada de Suplee, 2011. Assessment Methodology for Determining Wadeable Stream Impairment Due to Excess Nitrogen and Phosphorus Levels. Helena, MT: Montana Dept. of Environmental Quality.